

Investigating Event Nitrate Dynamics in Paired Vegetative Treatment Areas Receiving Silage Bunker Runoff Using a Simple Mixing Approach

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Abstract. Groundwater nitrate, often found in areas containing intensive agriculture, poses a human health risk at concentrations greater than 10 mg/L NO_3^- -N. Vegetative Treatment Areas (VTAs) are alternative treatment systems that have been proposed and implemented for the treatment of silage bunker runoff and other agricultural wastewaters on Concentrated Animal Feeding Operations (CAFOs) and other farms. The objective of this study was to temporally and spatially characterize runoff movement and nitrate dynamics on the surface and in the groundwater within paired VTAs treating silage bunker runoff following a precipitation event. A conservative tracer was applied prior to rainfall, and tracer and nitrate concentrations were then monitored in surface-water collector and monitoring well networks in the VTAs. Preferential movement of runoff was observed on the surface and to the groundwater in both VTAs. Nitrate dynamics were localized within the VTAs and did not correlate well with runoff movement. Nitrate concentrations not attributable to bunker runoff in excess of 20 mg/L NO_3^- -N were observed on the surface and in the shallow (61 cm) groundwater during the study. Observed groundwater nitrate concentrations were all less than 10 mg/L NO_3^- -N at a 162 cm depth.

Keywords. vegetative treatment, silage leachate, filter strip, silage bunker runoff

Introduction

Concentrated Animal Feeding Operations (CAFO) are common in the dairy industry. These facilities generate several production associated wastes that can have damaging environmental and health effects if not properly handled. It has been well documented that these wastewaters have high nutrient concentrations (Wright, 1996; Cumby et al. 1999; Cropper and DuPoldt, 1995). If improperly treated, these nutrients can cause groundwater impairment and eutrophication of surface waters. CAFOs, dairy and other types, across the country are required by the United States Environmental Protection Agency (USEPA) to control and treat these wastewater discharges. Collecting and storing all wastewaters for later field-spreading is cost-prohibitive for most producers; consequently, the practice of collecting wastewaters and passively distributing them for treatment by a vegetative treatment area (VTA) is common practice in New York and elsewhere (Wright et al., 1993). Because VTAs located on CAFOs receive large volumes of potentially polluting wastewaters, their treatment effectiveness is imperative.

Many of the previous studies on VTA effectiveness have focused on the removal of contaminants from surface water; primarily agricultural non-point (edge-of-field) runoff sources of relatively low contaminant concentrations (Dillaha et al., 1989; Magette et al., 1989; Lim et al., 1998; Abu-Zreig et al., 2003). The majority of studies that have been conducted on the treatment of concentrated waste streams have focused on feedlot runoff; Koelsch et al. (2006) provides a thorough review. In contrast, little attention has been given to silage bunker runoff (Wright et al., 2005). Due to the high concentration of organic compounds, undiluted silage leachate is a very polluting substance and can have a pH value of 4, BOD₅ concentrations in excess of 50,000 mg/L, 3,700 mg/L organic-nitrogen, an ammonia-nitrogen level of 700 mg/L, and over 500 mg/L of total phosphorus (Cropper and DuPoldt, 1995). The production of this waste stream and the associated treatment difficulties have increased in proportion with dairy farm expansion (Wright and Vanderstappen, 1994).

With such high concentrations of nitrogen forms, leaching of nitrate from VTAs into deeper groundwater is of special concern. The World Health Organization (WHO) recently reconfirmed that if levels are above 10 mg/L NO₃⁻-N, drinking water is unsafe for human consumption (WHO, 2004). It has been estimated that up to 26% of wells in intensive agricultural areas are contaminated with nitrate (Mueller et al., 1995). In the Northeast, nitrate contamination of groundwater can often be related to intensive agriculture and coarse-textured soils (Poe et al., 1998).

Little information exists regarding nitrogen removal from infiltrated water in VTAs in the Northeast. Schellinger and Clausen (1992) postulated that excessive hydraulic loading in a VTA in Vermont resulted in poor performance due to inadequate detention time, but did not address nitrogen-removal mechanisms. Yang et al. (1980) observed significant reduction in ammonium concentrations in the shallow groundwater below a VTA in Illinois, and attributed losses primarily to cation exchange complexes in the soil. Woodbury et al. (2005) attempted to monitor nitrogen movement 1.8 m beneath a VTA in Nebraska, but did not detect any percolation to that depth during a four year period. In Georgia, Hubbard et al. (2007) determined that a similar vegetated buffer system treating swine lagoon effluent at the farm-scale was effective at assimilating nitrogen and preventing leaching of nitrate over 10 mg/L NO₃⁻-N.

In general, the hydrology of VTAs can play a large role in the success of treatment mechanisms. For example, preferential flow paths on the surface of various types of vegetated filter strips have been widely observed and their impact on pollutant removal from surface water documented (Blanco-Canqui et al., 2006; Helmers et al., 2005; Dosskey et al., 2002). In

contrast, the degree to which preferential flow to the subsurface can impact solute transport to the groundwater and nitrogen transformations in VTAs has received limited attention. Kim et al. (2006) investigated both surface and sub-surface preferential flow paths and soluble reactive phosphorus (SRP) movement within a VTA for milkhouse wastewater treatment and found SRP removal was minimal within flow paths. Schellinger and Clausen (1992) partially attributed poor VTA treatment performance and rapid travel times in the subsurface (much shorter than those calculated using the Darcian velocity) to preferential flow. In natural systems, preferential flow paths have been known to rapidly transport event water and solutes laterally down-slope where it can mix with pre-event water (Waddington et al., 1993). Casey and Klaine (2001) also observed rapid mixing of pre-event and event water within a riparian wetland receiving golf course runoff, and saw different levels of nutrient attenuation between matrix and macropore flow (Casey et al., 2004). Tindall et al. (1995) observed vertical leaching of nitrate through macropores in soil columns and concluded that denitrification activity can be affected by preferential flow. The implications of subsurface preferential flow in VTAs for nitrate dynamics are generally unknown and deserve further attention as the installation of these systems becomes more widespread.

The objective of this study was to temporally and spatially characterize runoff movement and nitrate dynamics on the surface and in the groundwater within paired VTAs treating silage bunker runoff following a precipitation event.

Materials and Methods

Study Site

The study was conducted on a VTA system constructed in 2004 and put into operation in 2005 on a dairy farm in central New York, near the village of Freeville. The study site receives an average yearly precipitation of 93 cm and is located within the Fall Creek watershed. The farm is classified as a CAFO by the USEPA, and the VTA system was designed for the treatment of the farm's silage bunker storm runoff through complete infiltration of events less than the 24-hr, 25 yr storm (i.e. 11.7 cm). The VTA system is divided into two adjacent treatment areas (West and East), each having a slope of 3.3% and measuring 66 m long and 36 m wide. The treatment areas are planted in a mixture of reed canarygrass (*Phalaris arundinacea*), redtop (*Agrostis alba*), and tall fescue (*Festuca elatior*). The soil is classified as a Langford Channery silt loam (Fine-loamy, mixed, active, mesic Typic Fragiudepts), which consists of 40-70 cm of moderately permeable silt loam, underlain by a very dense, firm, slowly permeable silt loam restrictive layer (i.e. fragipan) (Soil Survey Staff, 2006). Each area is designed to receive half of the storm runoff from an 8900 m² concrete silage bunker, where both grass and maize ensilage is stored. Lower flow rates from the bunker, predominantly silage effluent during dry periods, are diverted and stored in an underground tank for later mixing with manure slurry. Storm runoff from the bunker passes through a series of coarse metal screens and then into a concrete settling basin, where it is divided and directed to the treatment areas via gravity flow through two underground 30.5 cm diameter pipes. Flow traveling to each treatment area is then discharged onto a level 90 cm wide concrete pad that spans the width of the top of the treatment area. A 3 meter wide berm, constructed of 7.6 to 15.2 cm diameter stone aggregate, separates the concrete pad from the vegetated area and is intended to aid in infiltration and uniform distribution of the flow as it passes onto the treatment area.

Instrumentation

Surface-water collectors for sampling surface water and monitoring wells for sampling subsurface water at two depths were installed within, upslope, and downslope of each treatment area. Each monitoring well network consists of a grid of three transects and five rows of well locations (Fig. 1). The labeling convention for the sampling points refers to the transect ('A', 'B', or 'C'), row number (Background or 1-4), and soil surface, or shallow or deep level in the profile. The 'Background' row is located upslope of the distribution trench and Row 4 is located downslope of the designated treatment area. Transects are spaced 9 m apart and rows are spaced 22 m apart. Transect B also contains a well location upslope (i.e. Background) and down-slope (i.e. Row 4) of the designed treatment areas. At every well location, a monitoring well at an approximate depth of 61 cm was installed. Surface-water collectors were just installed within the treatment areas (i.e. Row 1-3). Each well location in Transect B also contains a monitoring well at a 165 cm depth. The shallow monitoring well was installed so that the bottom was located at the interface of the restrictive layer and the overlaying soil. The monitoring wells in Transect B were constructed of 5.1 cm diameter PVC pipe and were installed in April 2006. The surface-water collectors and monitoring wells in Transects A and C were constructed of 3.8 cm diameter PVC pipe and were installed in August 2007. Monitoring wells were plugged on the bottom with a rubber stopper and had 1.15 cm openings extending from the bottom to a height of 25 cm. During installation, sand was placed between the perforated section and the surrounding soil, and a bentonite clay seal was placed on top of this sand to prevent the intrusion of surface water. Surface-water collectors were also plugged on the bottom, but have 1.15 cm openings starting at a 15 cm distance from the bottom and extending upward for 10 cm. These collectors were installed so that 5 cm of openings protruded above the soil surface and 5 cm of openings extended below the soil surface. Perforated sections on both types, wells and collectors, were wrapped with 10 mil thick polyester (Reemay) geo-synthetic fabric.

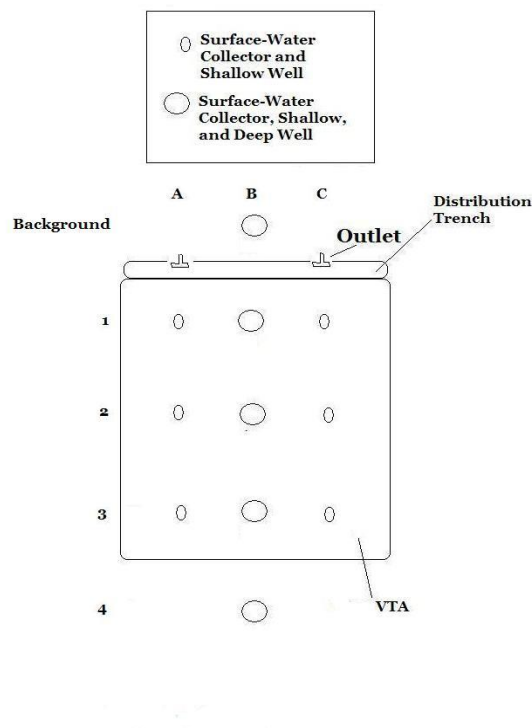


Figure 1: Monitoring network in treatment area

Rainfall was recorded at the study site at 5 minute intervals using a tipping-bucket rain gauge fitted with an event recorder (Spectrum Technologies, Inc. Watchdog Model 115). Water-level loggers (TruTrack, Ltd. WT-HR 1000) were installed in the shallow monitoring wells in each B transect on July 24, 2007, and groundwater levels were recorded at 10 minute intervals until the loggers were removed to prevent freezing on January 8, 2008. Stage measurements in the settling basin were recorded at 5 minute intervals using a compensated pressure transducer (Druck PDCR 830, 1 PSIG range) installed in a stilling well, and connected to a data recorder (Telog Instruments, Inc. R-2109). The circular PVC risers can be treated as weirs, and flow rates into each treatment area were calculated using the rectangular weir equation (Haan et al., 1994):

$$Q = CLH^{1.5} \quad (1)$$

Where Q is discharge in cubic meters per second, C is the weir coefficient, L is the circumference of the riser in meters, and H is the stage in meters. The weir coefficient was determined to be 1.66 through field calibration. Flow volumes were calculated by integrating flow rates over time during which runoff occurred. The East riser is slightly lower than the West riser within the settling basin; as a result, the East treatment area receives a greater runoff volume than the West treatment area.

Monthly sampling of the monitoring wells and periodic event sampling of the silage bunker runoff was performed for 22 months prior to the tracer study. These samples were analyzed for a suite of chemical constituents, including chloride, $\text{NH}_4^+\text{-N}$, and $\text{NO}_3^-\text{-N}$ concentrations (data not shown). Although silage effluent contains predominantly organic nitrogen, ammonium is the dominant form of inorganic nitrogen entering the treatment areas. The average concentration of sampled incoming $\text{NH}_4^+\text{-N}$ is 58.7 mg/L, while the average nitrate concentration is only 4.0 mg/L $\text{NO}_3^-\text{-N}$.

Soil Sampling

Soil samples were collected at each well location in Transect B in both treatment areas approximately three weeks before the tracer study (10/17/2007). Soils were sampled from 0-7, 7-30, 30-60, 60-90, 90-120, 120-150, and 150-180 cm. Samples were air dried, passed through a 2 mm sieve, and submitted to Cornell Nutrient Analysis Laboratory for Morgan extractable nitrate analysis (Morgan, 1941).

Tracer Study Procedure

Uncertain fragipan hydrology, evidence for preferential flow in similar systems, and isolated occurrences of significantly greater nitrate concentrations during the previous monitoring study motivated the initiation of an event study. This event tracer study was performed in November 2007. The use of conservative solute (e.g. chloride) data to investigate nitrogen transformations is common in similar systems (Altman and Parizek, 1995; Verchot et al., 1997; Mengis et al., 1999; Lowrance et al., 2000; Vellidis et al., 2003; Casey and Klaine, 2001; Duval and Hill, 2007). For this application, chloride was used as a non-adsorbing tracer to characterize flow and examine nitrate dynamics at the event scale within each treatment area. The tracer solution added to each treatment area was composed of 45.4 kg of 94-97% CaCl_2 (Scotwood Industries, Inc., USA) thoroughly mixed with 1140 L of tap water from Cornell University's Homer C. Thompson Vegetable Research Farm, resulting in an input Cl^- concentration of 13.8 g/L. The average input $\text{NO}_3^-\text{-N}$ concentration of the tracer solution was 0.45 mg/L. The tracer

solutions were added in each treatment area's (East and West) respective inlet in the settling basin. Additions were three hours apart, and were timed so that they would directly precede a predicted precipitation event. After the tracer additions, a rainfall event occurred within 5 hours and 3 hours of the East and West additions, respectively. Sampling of surface-water collectors and monitoring wells commenced on the East side within 3.5 hours of the rainfall event, and within 4.5 hours on the West side. Sampling then occurred once every 4 hours for 24 hours, and then once a day for seven days after the tracer addition. All surface-water collectors and monitoring wells were purged directly before the tracer additions and water was saved for analysis.

Water samples were collected in 240 mL plastic bottles using a vacuum pump. Bottles were then placed in a cooler and transported to the Soil and Water Laboratory at Cornell University where all water samples were vacuum-filtered through 0.45 µm filter within 24 hours of collection. The filtrate was stored at 4°C, and was analyzed within five days for Cl⁻ and NO₃⁻-N concentrations using ion chromatography (DIONEX, ION Pac[®]AS18).

Data Analysis

Chloride concentrations in shallow well water indicated that the water did not simply move uniformly from the distribution point down the slope to the lower end of the VTA. Chloride concentrations in samples were highly variable, both spatially and temporally. O'Donnell and Jones (2006), after observing similar non-uniformity in a riparian zone, utilized conservative solute data and a two end-member mixing model to determine contributions in groundwater from two distinct sources. A simple conceptual-based approach was similarly employed to provide an indication of how the tracer moved, assuming that samples from wells were essentially a mixture of runoff and existing groundwater. Likewise, samples from the surface-water collectors were simply assumed to be a mixture of runoff and rainwater. Thus, in order to calculate the relative contributions of each source in a sample at each sampling time, simple mixing equations were applied and solved simultaneously:

$$(Cl)^t_{well} = f^t_{gw}(Cl)_{gw} + f^t_{runoff}(Cl)_{runoff} \quad (2)$$

$$1 = f^t_{gw} + f^t_{runoff} \quad (3)$$

where $(Cl)^t_{well}$, $(Cl)_{gw}$, and $(Cl)_{runoff}$ are the concentrations of chloride in the well at sampling time, t , and in each source, either existing groundwater (gw) or runoff ($runoff$), respectively; f^t is the fraction of water derived from each source at each sampling time. This value serves as a simple indicator for tracer movement through the treatment areas. The chloride concentration measured directly before the tracer addition was used as the existing groundwater concentration for each location. For determination of the runoff chloride concentration for shallow and deep layer calculations, it was assumed that there was complete mixing of silage bunker runoff with the tracer solution in the distribution area above the stone berm, and then with rainfall on the treatment area upslope of a given row of monitoring wells. The silage bunker runoff chloride and nitrate concentration used for calculations was the average bunker runoff concentration over the 19 months prior to the tracer study. The rainfall chloride and nitrate concentrations were estimated using data from the National Atmospheric Data Program's (NADP) NY08 station (NADP, 2006). Predicted concentration of nitrate in groundwater at each sampling time was then calculated based on the f^t_{runoff} values:

$$P^t = (1-f_{runoff}^t)(NO_3-N)_{gw} + f_{runoff}^t(NO_3-N)_{runoff} \quad (4)$$

where P^t is the predicted concentration of nitrate in the monitoring well at each sampling time and (NO_3-N) is the nitrate concentration in each source. Nitrate runoff and existing groundwater concentrations were determined analogous to chloride concentrations. As a conservative assumption and simplification, if the f_{runoff}^t value was negative, a value of zero was substituted to indicate that there was no water originating from runoff in that sample.

Similar calculations were also performed with reference to surface-water samples, with the exception that uniform mixing with rainfall was not assumed, but was determined using (2) and (3) and substituting rainfall for groundwater concentrations. Likewise, for the calculation of P^t in surface samples, the concentration of NO_3^- -N in rainfall was substituted for the concentration in groundwater.

Saturated hydraulic conductivity was determined in the shallow layer within the B transect in each row using a slug test procedure (Bouwer and Rice, 1976). Expected pore water velocities within the treatment areas were then determined using Darcy's Law:

$$v = sK/n \quad (5)$$

where v is pore water velocity, s is hydraulic gradient, K is saturated hydraulic conductivity, and n is porosity.

Results and Discussion

Hydrology

Average hydraulic parameters for the soils in each treatment area are listed in Table 1. K values are lower than typical values (0.037-0.122 m/day) for this soil type (Soil Survey Staff, 2006). It is possible that the lower K values are a result of soil disturbance and compaction during construction. Melvin and Lorimor (2007) observed reduced infiltration rates and hydraulic conductivity in recently constructed VTAs in Iowa. Shallow groundwater was generally elevated into the root-zone within the treatment areas, possibly due to flow-restricting characteristics of the fragipan.

Table 1: Average hydraulic parameters within treatment areas

Treatment Area	n	K (m/day)	v (m/day)
West	0.5	0.0072	0.00085
East	0.5	0.014	0.0017

Event rainfall and silage bunker runoff are displayed in Figure 2, along with the time of tracer addition and the commencement of sampling. The farm received a total of 7.8 mm of rainfall during the tracer study. Initially, 1.5 mm of rainfall occurred directly following the East tracer addition, another 5.3 mm began four hours later, and then another 1 mm of rain fell approximately four hours after that, directly preceding sampling. The East and West treatment areas received a total of 40.5 m³ and 36.0 m³ water, respectively; volumes include tracer solution, silage bunker runoff, and direct rainfall. The observed arrival times of tracer concentration peaks are significantly less than the corresponding travel times determined using

the calculated pore water velocity. The arrival of peak concentrations occurred within the first 24 hours after the storm in all well locations; whereas, calculated travel times are all greater by four orders of magnitude or more. Within the treatment areas, the hydraulic gradient indicates that groundwater movement was generally down the slope, away from the distribution trenches, and slightly towards the West area, with the exception of Row 4 (Table 2). Water-level logger data also indicate that before the study, the water table was much closer to the surface in the East area than in the West area, and becomes deeper when moving from Row 1 to 4, indicating a ‘mounding’ of the groundwater beneath the treatment area.

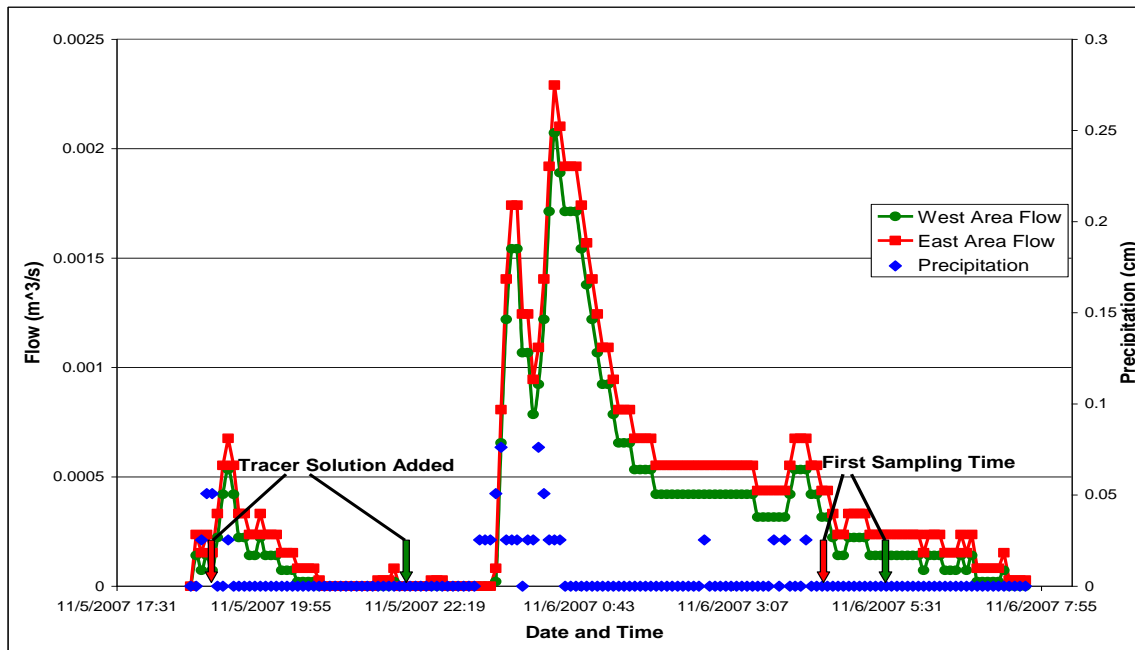


Figure 2: Five minute precipitation and flow to treatment areas

Table 2: Shallow groundwater table (GWT) before tracer study

Row	West Area			East Area		
	GWT Elev. (m)	Ground Elev. (m)	Depth from surface (m)	GWT Elev. (m)	Ground Elev. (m)	Depth from surface (m)
1	352.33	352.44	0.11	352.45	352.47	0.02
2	351.18	351.39	0.21	351.21	351.29	0.08
3	349.76	350.23	0.48	349.93	350.01	0.08
4	348.80	349.26	0.46	348.25	348.72	0.47

Soil Nitrate

Results from soil analyses are presented in Figure 3. Little nitrate is present in most locations beneath the 30-60 cm depth. Row 4 in the East treatment area exhibits consistently elevated soil nitrate persisting down into the 60-90 cm depth. The same is true for the West Background location, where the soil nitrate is also elevated above values within the treatment area itself; a possible result of drier conditions and little denitrification activity. Row 3 in the East area exhibits an extremely high concentration of nitrate present in the 0-7 cm soil depth.

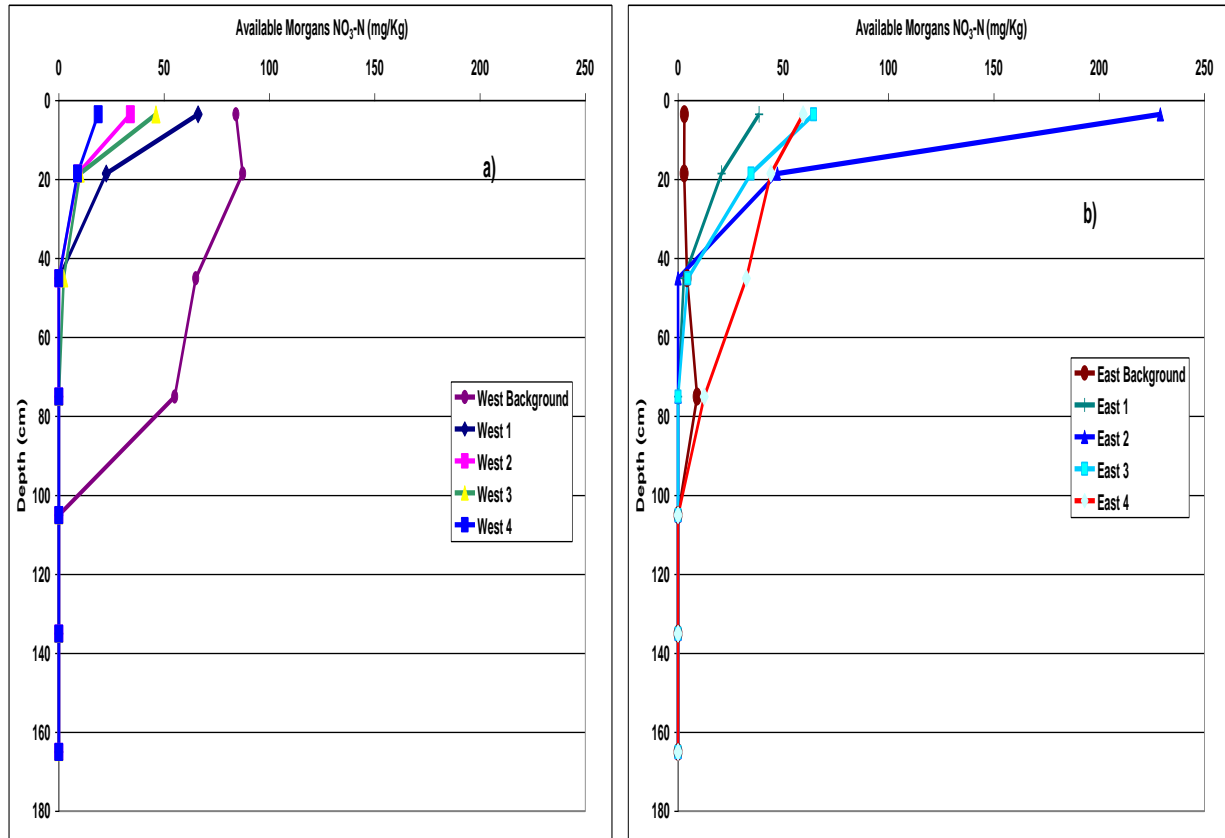


Figure 3: Soil nitrate in West (a) and East (b) areas (10/17/2007)

Tracer Movement

Surface

In the West area (Fig. 4, left), where the water table was initially further from the soil surface, the few samples of surface flow are confined to Transects A and C, and Rows 1 and 2. No runoff is sampled on the surface in Transect B or Row 3, and flow generally bypasses sampling points in Row 1, except for a single sample at 7.5 hours in Transect C. Water ceases to be available for sampling after 19.5 hours.

Surface flow appears to be well distributed and persists longer in the East area (Fig. 4, right). Fractions tend to show peak amounts of runoff in samples throughout the first day. Peak values occurred at the first sampling time (9.5 hrs) in two locations, at 13.5 hours in five locations, and at 17.5 in one location. Runoff remains present in collectors in two locations (i.e. Row 1, Transect C and Row 2, Transect A) throughout the study period. Furthermore, in the Row 1, Transect C location, the fraction of runoff present continued to increase throughout the study period. Compared to the surface-water collectors in the West treatment area, water was more often present for sampling from the collectors in the East area. This was likely due to a greater volume of received runoff during the event, the initial water table being much closer to the surface in the East area (e.g. 2 cm in East Row 1), and prevalent concentrated flow paths. These factors likely resulted in rapid saturation of the entire soil profile upslope of Row 2 as well as soils underlying concentrated flow paths, preventing infiltration of a considerable portion of tracer and bunker-runoff and augmenting surface transport. Such concentrated flow paths are often noted in these systems, and were visually observed in this study. The high fractions of

runoff observed in surface samples in Row 3 through the first day suggest that there was a surface effluent from the East treatment area. Although no surface-water collectors were installed in the cornfield below this treatment area, surface flow leaving the area was observed during the study. It is postulated that complete soil saturation also augmented the persistence of surface flow in the two locations in the East area that are sampled throughout the entire study. This is likely a result of some surface flow attenuation within the stone berm and near-surface soil, and subsequent slow surface/near-surface lateral transport across/through saturated soils via established concentrated flow paths.

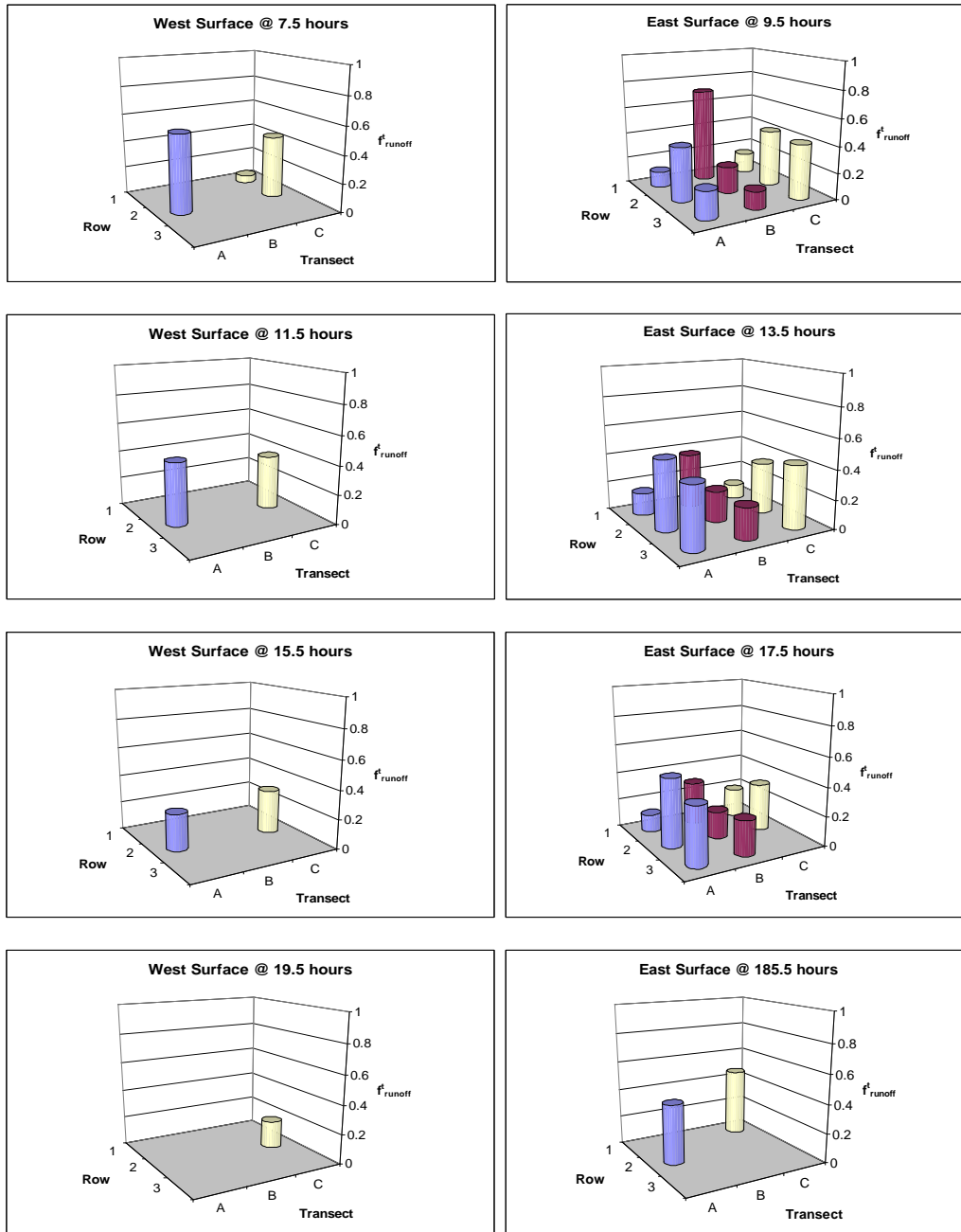


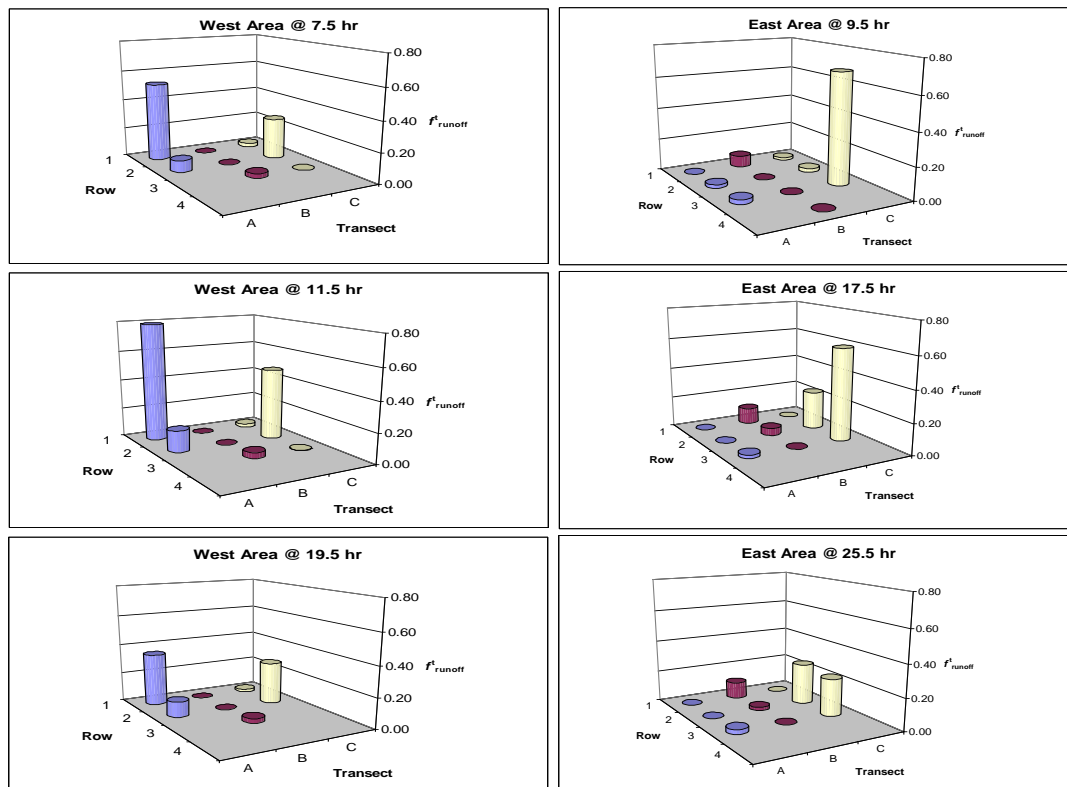
Figure 4: Runoff movement on surface at selected sampling times

Shallow

The fractions of water originating from runoff in the shallow layer at selected sampling times are shown in Figure 5. In general, values indicate that runoff did not move uniformly down-slope through the shallow soil. In the West area, the runoff is predominantly detected in Transects A and C; observations indicate that little runoff entered the shallow layer of Transect B (max. = 0.04). No water was available for sampling in Row 4 during the sampling period. The maximum estimated fraction of runoff (0.79) in this treatment area occurred in Row 1, Transect A at a sampling time of 11.5 hr, but very little runoff was detected in Transect C of the same row. The maximum fraction of runoff observed in Transect C (0.48) occurred in Row 2, at a time of 11.5 hr. At 110.5 hours, only five wells still contain water, and all fractions are less than 0.06. Water continues to drain from the shallow layer, and fractions continue to decrease to less than 0.03 at 183.5 hours.

In the East area, runoff was predominantly detected in Transect B and C; observations indicate that little runoff entered the shallow layer of Transect A (max. = 0.04). Water was only available for sampling twice during the study period in Row 4 (9.5 and 43 hrs). The maximum estimated fraction of runoff (0.69) in this treatment area occurred in Row 3, Transect C, at a sampling time of 9.6 hours. The fraction of runoff in Transect B did not exceed a value of 0.12 in any row. The fraction of runoff in Row 2 of Transect C peaks after the peak in Row 3, and no definitive peak was observed in Row 1. Water remains present for sampling in eight locations through the study period, but fractions are all less than 0.10.

These results indicate that preferential flow occurred within the treatment areas, as numerous locations showed significant mixing of event water with pre-event water well before predicted travel times. This transport is most likely a combination of both subsurface and surface preferential flow, and has potential implications for transport of solutes laterally downslope and vertically downward into groundwater.



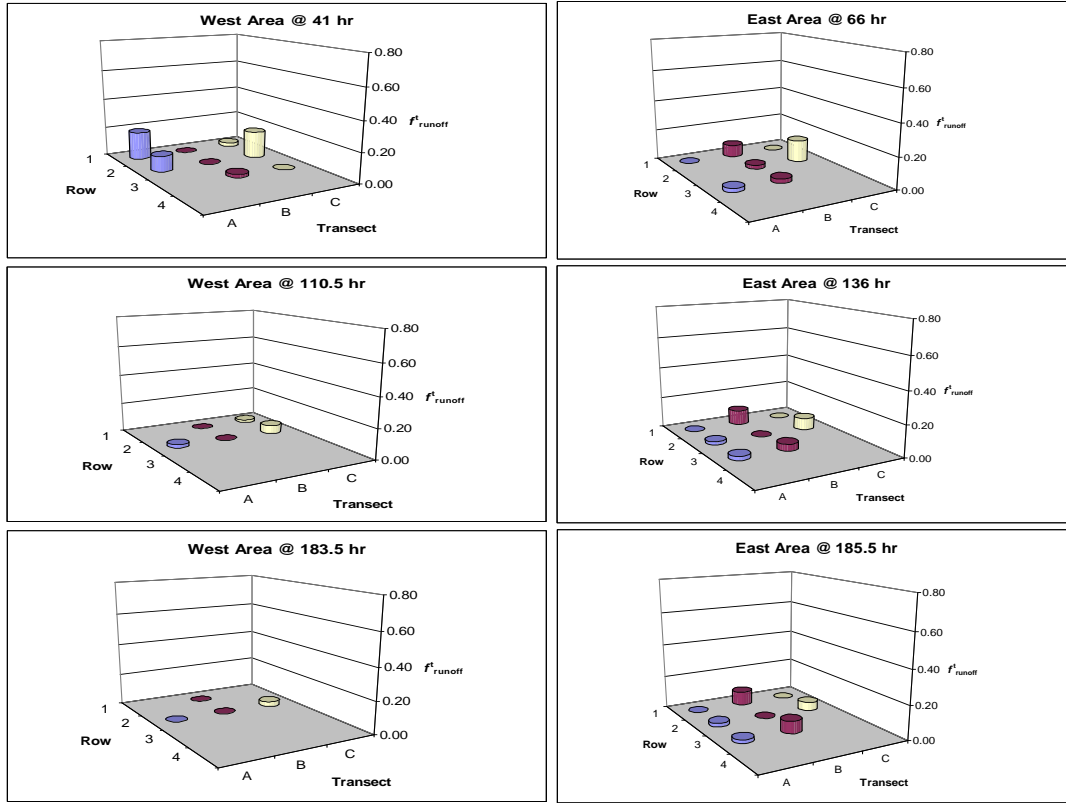


Figure 5: Runoff movement in shallow layer at selected sampling times

Deep

The fraction of water originating from runoff was similarly calculated for the deep monitoring wells (Transect B). In the West area (Fig. 6, left), observations indicate that very little, if any, runoff reached the deep layer during the course of the study, as values were all less than 0.01.

In contrast, observations in the East area (Fig. 6, right) tend to indicate that a very small portion of runoff did appear in the deep layer. Fractions in Row 4 are the largest, with a maximum value of 0.07 at a sampling time of 17.5 hours. This suggests that some surface runoff moved rapidly down through the shallow layer in Row 4, where no water table was present, and into the deeper layer. Fractions in Rows 1, 2, and 3 are all less than 0.05, and generally indicate little runoff present.

Even though significant runoff did reach the shallow groundwater, results indicate that there was very little transport of runoff through the fragipan and into deeper groundwater during the study period.

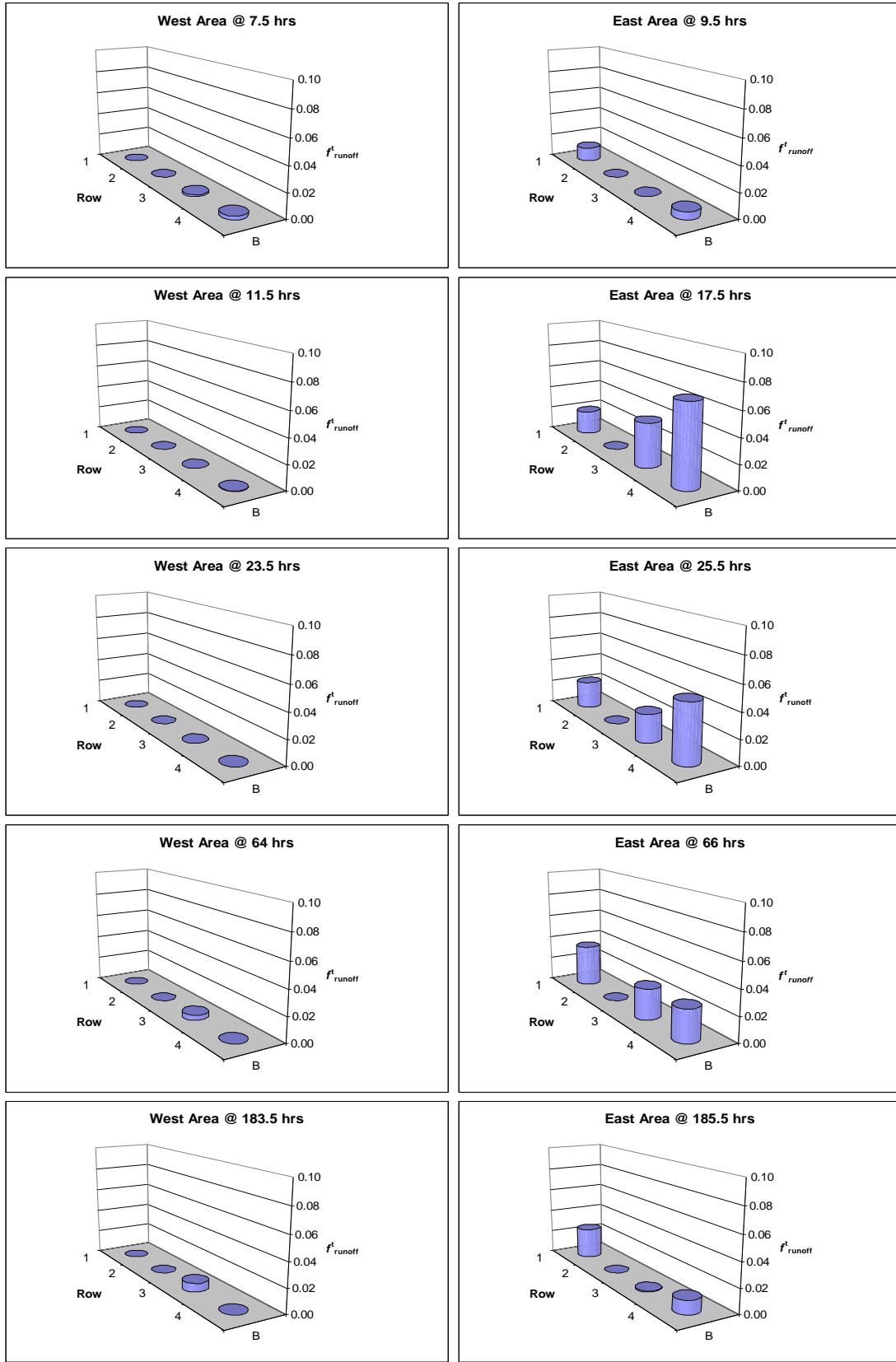


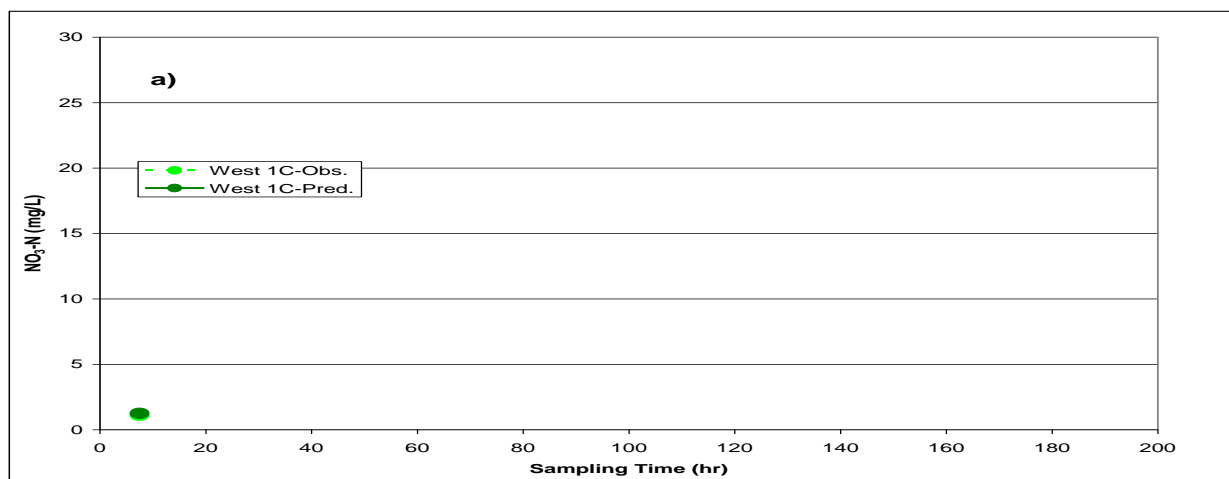
Figure 6: Runoff movement in deep layer at selected sampling times

Nitrate Dynamics

Results of the analysis of observed and predicted nitrate concentrations help indicate the degree of net nitrate retention or production, while accounting for the effects of dilution. Here, 'production' refers simply to the net increase in nitrate concentration. Possible sources of nitrate production include mineralization of organic nitrogen followed by nitrification, nitrification of existing ammonium, and leaching of existing nitrate from the soil water into the sampled groundwater. Organic nitrogen mineralization and subsequent nitrification is likely to be negligible during the duration of this tracer study, but nitrification of incoming or existing ammonium may occur rapidly with wastewater applied to the soil. Likewise, 'retention' includes any mechanism that results in a net reduction of nitrate concentration. Possible mechanisms of nitrate retention include denitrification and plant uptake. Plant uptake is typically expected to be a significant process affecting nitrate removal in VTAs, (USDA, 2006), and cannot be eliminated as a nitrate sink. However, given the duration and seasonality of the study, its contribution to nitrate removal is expected to be minimal. Furthermore, possible leaching of soil-water nitrate into the groundwater is not separately accounted for, and cannot be eliminated as a source of nitrate production. If leaching of nitrate from overlying soil contributed significantly to nitrate production, it is expected that this addition would have occurred simultaneously with tracer arrival, as incoming water moved downward into the saturated zone.

Surface

On the surface in the West area (Fig. 7), relatively small differences in predicted and observed nitrate concentrations were observed. In Row 2, the initial nitrate concentration in Transect C was higher than predicted, but lower than predicted in Transect A. Also in Transect C, concentrations were predicted to consistently decrease through the first day, but observations indicate that they increased until 15.5 hr, and then decreased to within 0.11 mg/L of the initial concentration. Overall, calculations indicate that the net change was less than 5 mg/L NO_3^- -N in any location.



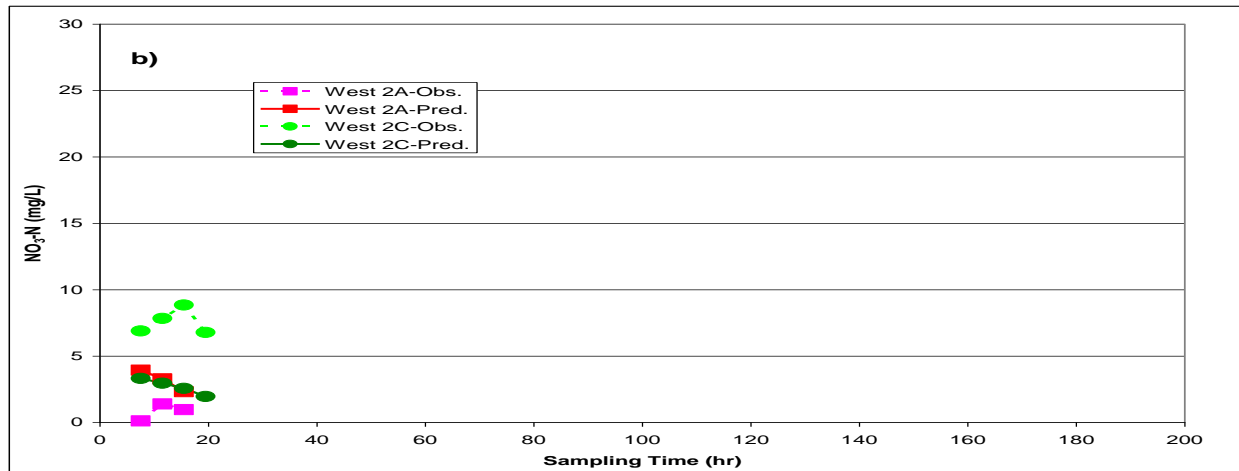


Figure 7: Observed and predicted nitrate on West surface Row 1 (a) and Row 2 (b)

The nitrate on the surface of the East area (Fig. 8) is generally low, with a few exceptions, including concentrations reaching 28 mg/L NO_3^- -N in Row 3. In Rows 1-3, Transect B, and Row 3, Transect A, the initial observed concentration was notably higher than predicted, indicating a production of nitrate. After the initial observed nitrate concentration, values decreased. This apparent drastic production could possibly be due to a lateral flushing down gradient and mixing of nitrate present in the surface soils with surface runoff. The following apparent consumption of nitrate would then be observed as this initial flush of nitrate moved out of the system. Tracer data indicating ample surface flow, and recent soil data showing that soil nitrate was considerably elevated in the surface soils of Row 2 in the East area (Fig. 3), support the postulation of lateral flushing across the surface of Row 3. Furthermore, the relatively high nitrate concentrations present in surface runoff in Row 3, at the lower end of the treatment area, indicate that surface export of nitrate from the East area is possible in similar sized events if flushing is indeed responsible for these high nitrate concentrations. Surface flow, and nitrate export, is likely augmented by a higher hydraulic loading, the resulting shallower groundwater, and a more rapid initiation of saturation-excess runoff.

An alternate explanation for this nitrate production is a rapid increase in nitrification activity following the addition of the ammonium-rich runoff to the soil. Assuming this was a controlling mechanism, the subsequent decrease in nitrate could then be due to denitrification coupled tightly to nitrification at the unsaturated-saturated soil interface. Additions of liquid swine waste high in ammonium have been reported to increase coupled nitrification-denitrification activity in soils (Fischer and Whalen, 2005). Nielsen and Revsbech (1998) also observed localized (in so-called 'hot-spots') coupling of nitrification-denitrification after waste application where interfaces existed between aerobic-anaerobic conditions in soils. As the groundwater in this treatment area is consistently shallow, including directly before the tracer event, it is likely that an aerobic-anaerobic interface did exist within the zone sampled by the surface-water collectors.

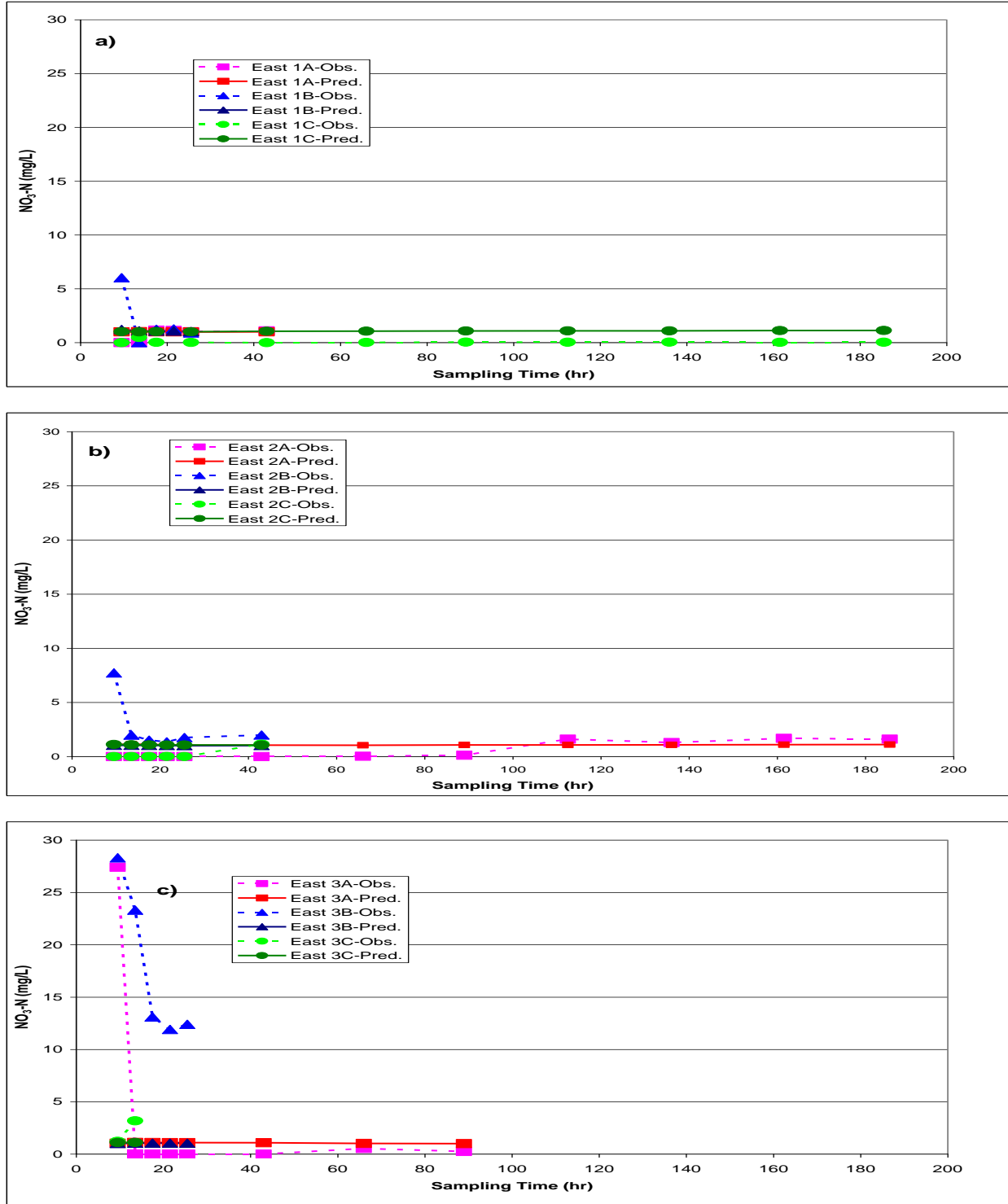


Figure 8: Observed and predicted nitrate on East surface Row 1 (a), Row 2 (b), and Row 3 (c)

The fractions of runoff present in surface samples (i.e. f_{runoff}) show little correlation with differences between observed and predicted nitrate concentrations (West correlation coefficient = 0.27; East correlation coefficient = 0.19). This suggests that preferential flow paths and general movement of runoff to some locations more than others on the surface of the VTA did

not singly significantly influence nitrate production or consumption processes. Areas of production and consumption were localized, and it is likely that spatial variation is influenced by host of factors. These factors likely include surface hydraulic flow paths (influencing degree of saturation and incoming substrates) and heterogeneity in organic carbon and microbial populations needed for biological processes.

Shallow

In the shallow groundwater in the West area (Fig. 9), where a significant fraction of runoff was present in Row 1, Transect A, but no surface flow was measured, there was little difference between the initial predicted and observed nitrate concentrations. In the next four hours however, predicted nitrate increased to over 4 mg/L, while observed concentrations decreased to close to 0 mg/L. This indicates that incoming nitrate was rapidly consumed in this location. In Row 2, Transect A and C both exhibit spikes in observed concentrations during the first day that were not predicted, 6.62 and 10.68 mg/L, respectively. Observed values in both locations then decrease, and are similar to predicted values by the second day. Both of these locations received relatively high amounts of runoff in the shallow layer and are also located within a surface flow path (Fig. 4), presenting the possibility of leaching through preferential flow paths. The addition of new water and accompanying oxygen and substrates also has the potential to influence the activity of microbial nitrate production/consumption processes that previously could have been limited; therefore, rapid nitrification cannot be eliminated as a production mechanism. Nitrate did not accumulate in Row 2 however, and was immediately consumed. In contrast, the observed concentration in Row 1, Transect B, fluctuated significantly during the study, but no runoff was measured on the surface or in the shallow groundwater in this location. The initial predicted nitrate concentration in this location was higher than in all other locations, and was not expected to vary throughout the study based on conservative tracer data. The initial observation was within 2.5 mg/L of the predicted value, but there was then a consumption of over 7 mg/L of nitrate during the first day, followed by a consistent production of 17.5 mg/L of nitrate during the next week. This discrepancy between predicted and observed values, accompanied by a lack of evidence for bunker-runoff intrusion, indicate that fairly rapid nitrate consumption and production processes can occur in localized areas and are not necessarily mediated by the addition of new water from the bunker. New water from direct rainfall on the VTA would have been present across the treatment areas, and could have potentially contributed to the nitrate production if it leached nitrogen species from the upper soil. The relatively high initial nitrate concentration, in conjunction with the production during the study, indicated that conditions for nitrate production were favorable in this location. In all other locations, observed nitrate concentrations were well under 5 mg/L after the first day, indicating that significant leaching of nitrate (i.e. concentrations approaching 10 mg/L) into the shallow groundwater, or microbial production within the shallow layer, was not widespread during the studied event in the West area.

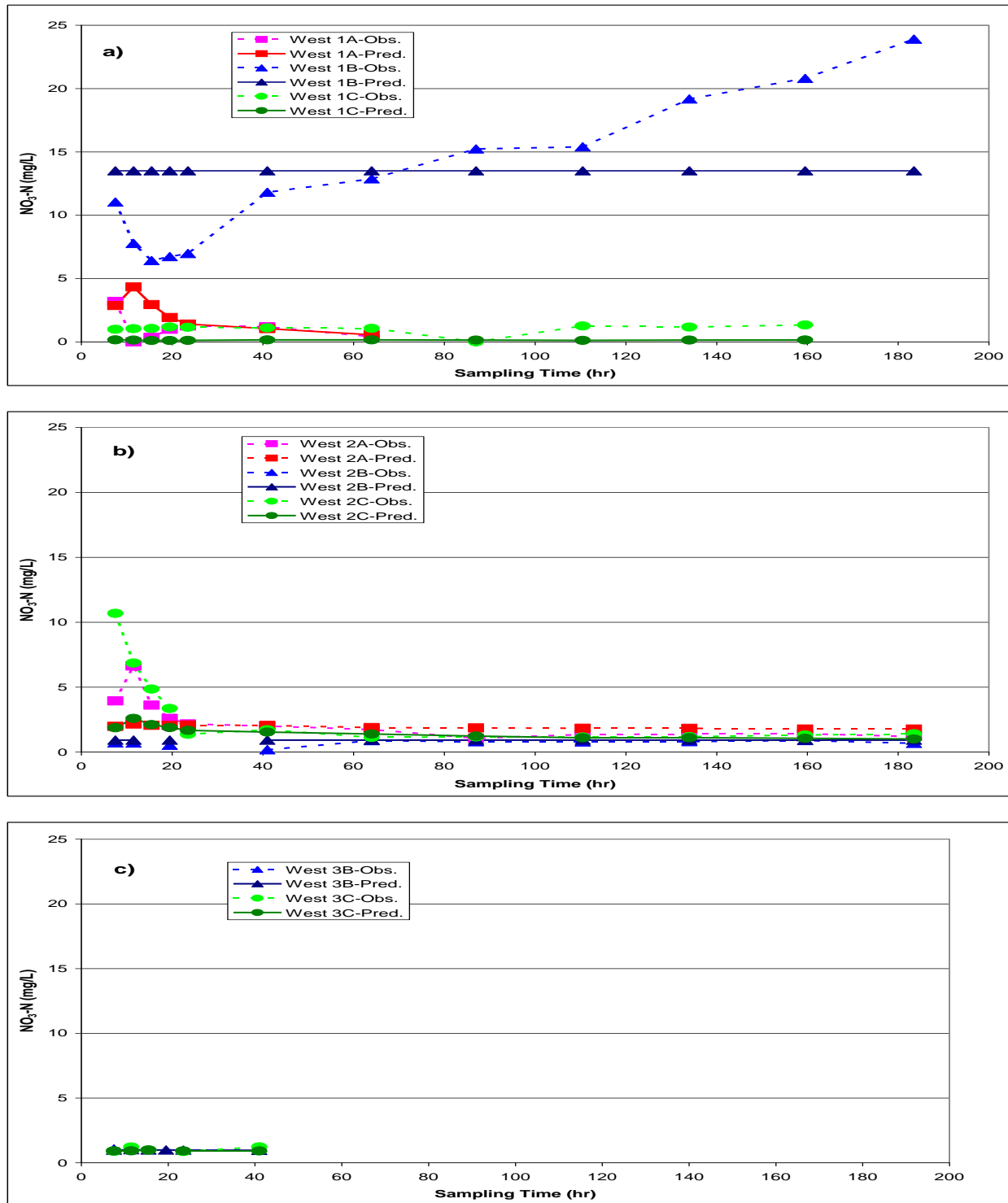
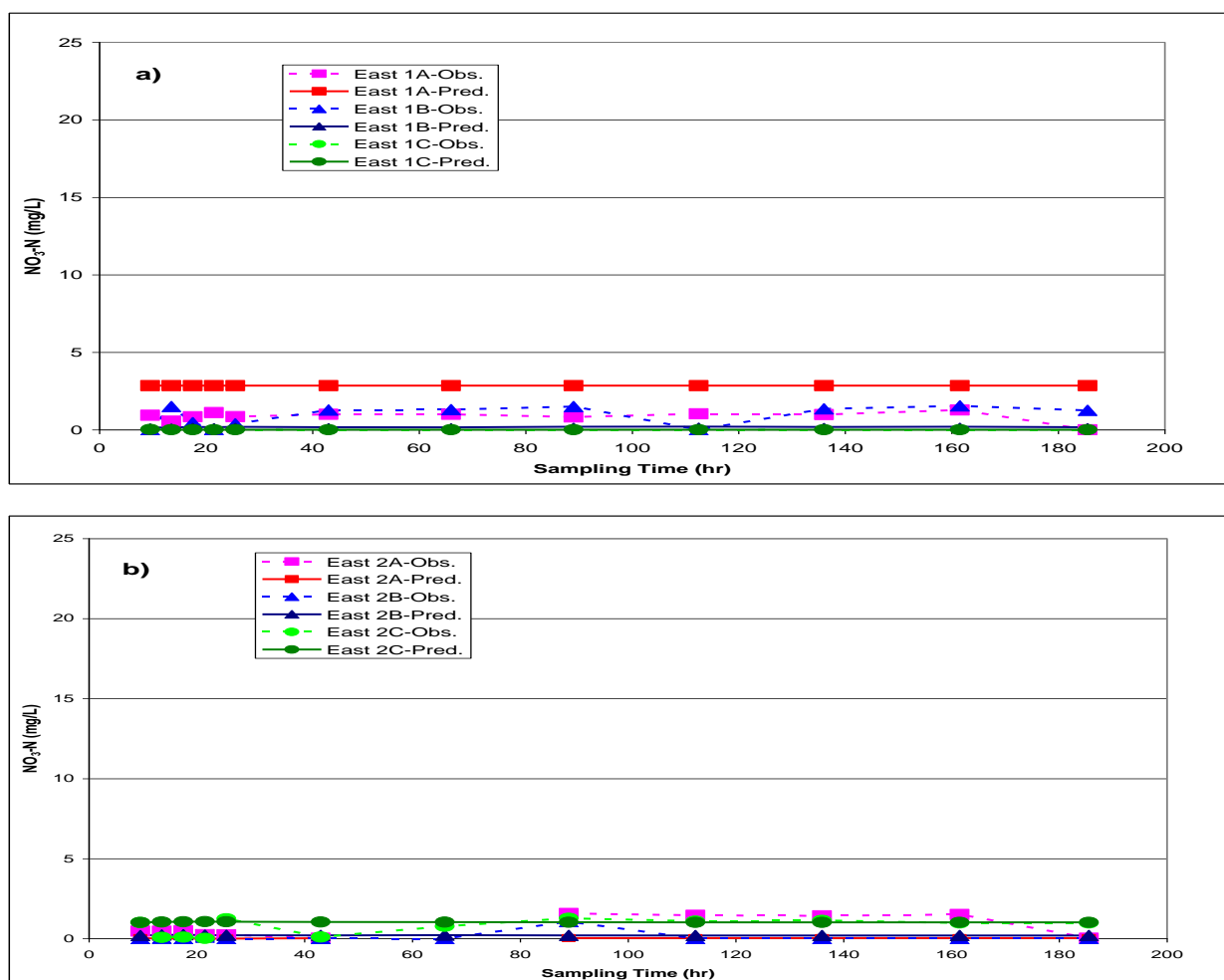


Figure 9: Observed and predicted nitrate in shallow layer in West area; Row 1 (a), Row 2 (b), and Row 3 (c)

Although significant runoff fractions were observed in a few locations within the shallow layer in the East area, differences between observed and predicted nitrate concentrations were minimal during the course of the tracer study in those, and all other, locations (Fig. 10). This suggests

that, even given the historically high incoming ammonium concentrations and relatively high levels of soil nitrate, nitrate production/consumption processes, including leaching, were minimal in the shallow layer of this area during the study period. It is postulated that the consistently higher hydraulic loading to this area, and resulting shallow groundwater, combined with ample organic carbon, aided in the formation of increasingly reduced conditions that have the potential to limit significant nitrification in the shallow layer. These conditions, which were likely to be more reduced than in the West area, would likely inhibit what could have been rapid nitrification observed in the shallow layer in the West area with the addition of event water. These reduced conditions and shallower groundwater possibly also reduced the production of nitrate prior to the event in the unsaturated zone, which was considerably less thick than in the West area, further reducing the chances of rapid nitrate leaching.



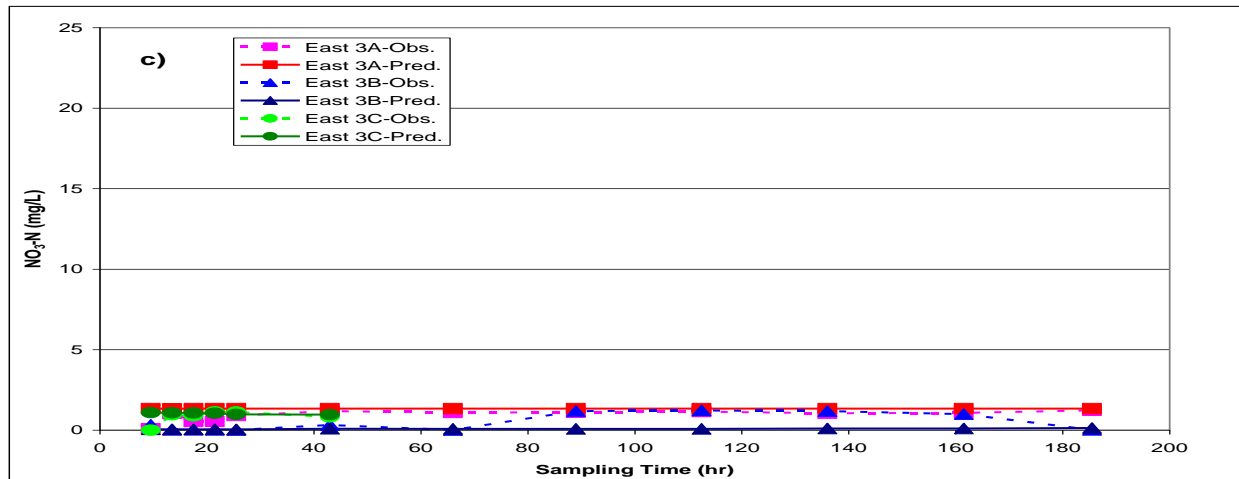


Figure 10: Observed and predicted nitrate in shallow layer in East area; Row 1 (a), Row 2 (b), and Row 3 (c)

Even though nitrate production is observed in two locations in the West area where significant runoff intrusion was measured, overall the fractions of runoff present in shallow samples (i.e. f_{runoff}) show little correlation with differences between observed and predicted nitrate concentrations (West correlation coefficient = -0.04; East correlation coefficient = -0.05). This suggests that, in general, preferential flow paths carrying surface water into the shallow layer of the VTA did not influence nitrate production or consumption processes. Although correlation coefficients indicate little influence, nitrate dynamics in Row 2, Transect A and C, cannot be ignored. Even though nitrate spikes and apparent preferential flow were observed in these locations, the nitrate that was produced (via leaching or other processes) was subsequently consumed within 24 hours. That consumption, along with the production of nitrate in Row 1, Transect B that was unaccompanied by preferential flow, suggests that nitrate in shallow groundwater is not significantly impacted at the event scale in the studied VTAs.

Deep

Although significant changes in nitrate concentrations are observed in locations in the shallow layer of the West area, no observed nitrate concentration fluctuations of note are recorded in the West deep layer (Fig. 11). Predicted nitrate concentrations also remain constant throughout the study period, indicating no nitrate production attributable to nitrate in runoff. All observed nitrate concentrations are very low, below 2 mg/L, and are generally within 1 mg/L of predicted concentrations. The absence of runoff in the deep layer and the lack of an observed increase in nitrate concentrations, especially in locations exhibiting nitrate production in the corresponding shallow layer, indicate that nitrate was not leached or produced otherwise in the deep layer during the study period.

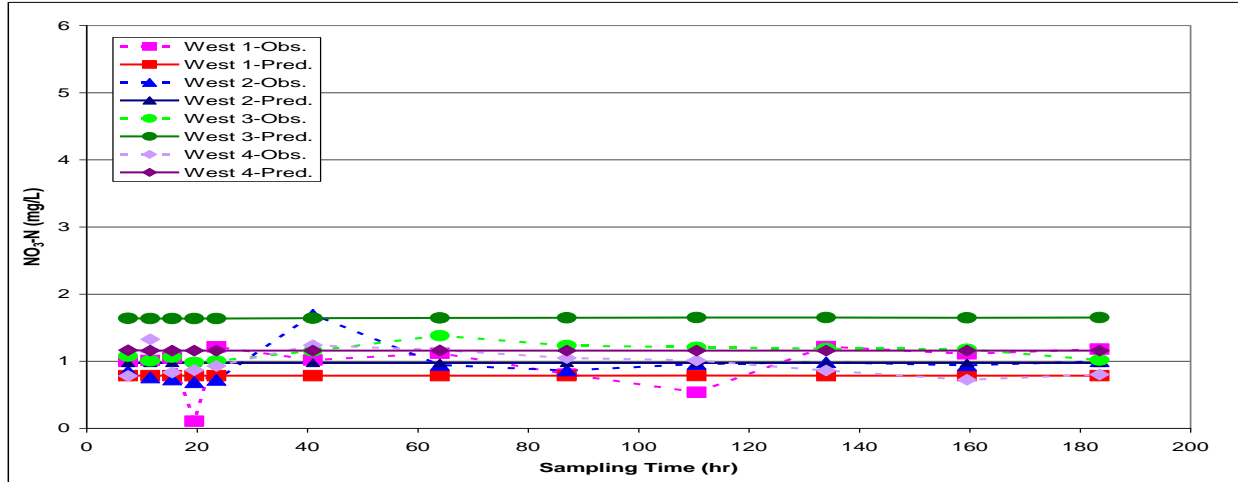


Figure 11: Observed and predicted nitrate in deep layer in West area

Converse to the West area, little variability was observed in nitrate concentrations in the shallow layer in the East area, but an unpredicted spike in nitrate was observed in the deep layer, along with generally greater observed concentrations than predicted, with the exception of Row 1 (Fig. 12). The spike (> 5 mg/L) occurred in Row 3 at the second sampling time, but was immediately followed by a decrease to below 2 mg/L. Predicted nitrate in this row never increased above 0.05 mg/L, indicating that the observed nitrate was not a result of nitrate in runoff from the bunker. Although tracer data indicated that some runoff was possibly present in the deep layer in Row 3, the fraction was very low, and other larger fractions were observed in other locations with no observed nitrate production. The spike did occur during the same time that observed nitrate was spiking on the surface at this location. It is possible that nitrate produced on the surface was preferentially transported downward to the sampling zone of the deep well between the well casing and the surrounding soil.

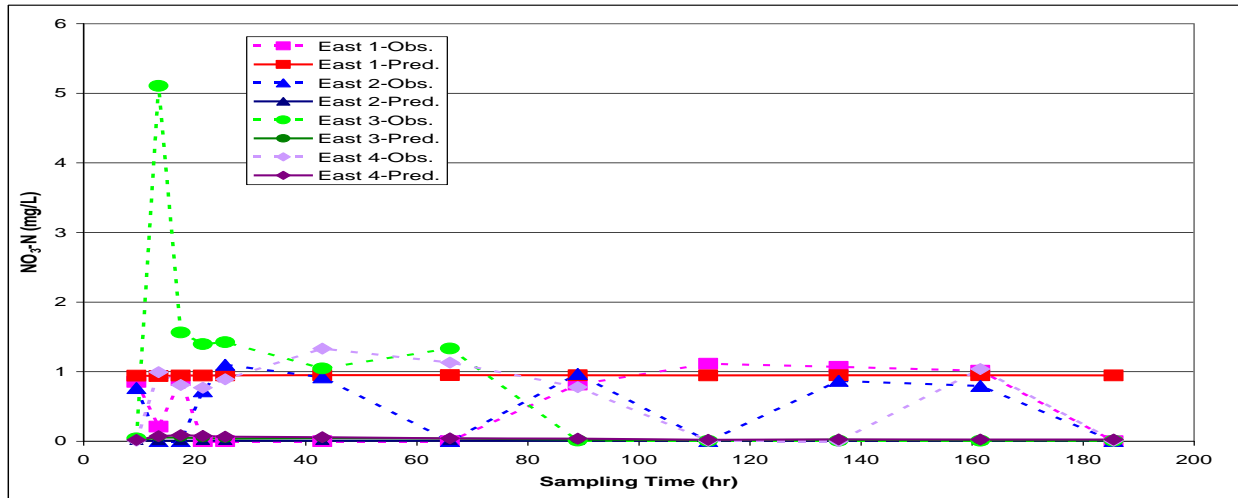


Figure 12: Observed and predicted nitrate in deep layer in East area

The fractions of runoff present in deep samples (i.e. f_{runoff}) show little correlation with differences between observed and predicted nitrate concentrations (West correlation coefficient = 0.09; East correlation coefficient = -0.21). This suggests that, in general, what little runoff that did move into the deeper layer of the VTA did not influence nitrate production or consumption

processes, including leaching. Although deep wells were not located in Transect A and C, the results and discussion above were generalized for the entire treatment area. Significant preferential flow was also not observed in the shallow layer in Transect B in either treatment area. Deep monitoring wells installed in Transect A and C would help better determine if preferential flow to the shallow layer affects nitrate dynamics in the deeper groundwater.

Conclusion

In conclusion, concentrated flow paths were observed on the surface of both treatment areas. A shallower groundwater table in the East area helped facilitate greater surface flow than in the West area. Surface effluent from the East area was visually observed during the study, indicating VTA failed to infiltrate all incoming runoff. Localized areas of significant nitrate production and consumption were observed on the surface of the treatment areas. Preferential transport to the shallow layer was observed within, and was similar between, the two areas. Little evidence for transport of incoming runoff through the fragipan to the deep layer was observed during the study period. Nitrate production and consumption is localized within the groundwater in these VTAs, and reached greater magnitudes and exhibited greater variability in the shallow West area, but, in general, returned to background levels within a day after the precipitation event. Increases in groundwater nitrate in the shallow and deep groundwater cannot be exclusively attributed to leaching by preferential flow, or rapid microbial processes, and were observed both in the presence, and absence, of new water. Groundwater nitrate concentrations in excess of the USEPA drinking water standard (i.e. 10 mg/L NO₃⁻-N) were observed at the 61 cm depth in the West VTA, but not at the 165 cm depth during a tracer study in two VTAs receiving silage bunker runoff.

When presented with such spatial and temporal variability in tracer movement, the mixing approach employed served as a useful tool that was simple to apply. Assumptions exist within this approach, but with limited resources reasonable estimations of solute dynamics can be made.

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